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Prescribed fires as ecological surrogates for wildfires: A stream and riparian perspective

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ABSTRACT

Forest managers use prescribed fire to reduce wildfire risk and to provide resource benefits, yet little information is available on whether prescribed fires can function as ecological surrogates for wildfire in fire-prone landscapes. Information on impacts and benefits of this management tool on stream and riparian ecosystems is particularly lacking. We used a beyond-BACI (Before, After, Control, Impact) design to investigate the effects of a prescribed fire on a stream ecosystem and compared these findings to similar data collected after wildfire. For 3 years after prescribed fire treatment, we found no detectable changes in periphyton, macroinvertebrates, amphibians, fish, and riparian and stream habitats compared to data collected over the same time period in four unburned reference streams. Based on changes in fuels, plant and litter cover, and tree scorching, this prescribed fire was typical of those being implemented in ponderosa pine forests throughout the western U.S. However, we found that the extent and severity of riparian vegetation burned was substantially lower after prescribed fire compared to nearby wildfires. The early-season prescribed fire did not mimic the riparian or in-stream ecological effects observed following a nearby wildfire, even in catchments with burn extents similar to the prescribed fire. Little information exists on the effects of long-term fire exclusion from riparian forests, but a "prescribed fire regime" of repeatedly burning upland forests while excluding fire in adjacent riparian forests may eliminate an important natural disturbance from riparian and stream habitats.

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1. Introduction

Wildfires result in a cascade of environmental changes in riparian and stream ecosystems in temperate forests of the western U.S. (Bisson et al., 2003; Everett et al., 2003; Gresswell, 1999; Minshall, 2003; Mellon et al., 2008; Pettit and Naiman, 2007; Power and Dietrich, 2002). Changing habitat conditions can influence community composition, instream energy production, and food web interactions that redirect energy flow towards higher instream trophic levels including predatory macroinvertebrates and salmonids (Minshall, 2003; Heck, 2007; Rieman and Clayton, 1997; Wootton et al., 1996). Post-fire erosion, snag-fall, and debris flows can supply important structural elements to stream channels, such as pool-forming large woody debris and gravel used in spawning redds (Knutson and Nae, 1997). Mosaic patterns of burn severity and extent can increase habitat heterogeneity and biodiversity in streams and riparian forests (Reeves et al., 1995; Swanson and Lienkaemper, 1978).

To avoid the detrimental effects of large, high severity fires and to restore fire disturbance patterns in western forest ecosystems, prescribed fire and mechanical forest thinning are being used as management tools, particularly near wildland-urban interfaces. The management goal of most prescribed fires is to reduce surface and ladder fuels to prevent stand-replacing crown fires that may threaten property and natural resources. Secondarily, prescribed fires are used to restore fire-dependent processes, ecosystems, and specific flora and fauna to forests that once burned frequently under mixed-severity fire regimes (Bruce, 1947; Friederichi, 2003). As such, prescribed fire is used as an ecological surrogate for wildfire, especially as wildfire is replaced by prescribed fire on the landscape (Boerner et al., 2008). While the effects of fire surrogates on fuel reduction, soils and vegetation structure have been examined in some communities (Boerner et al., 2009; Schwilk et al., 2009; Stephens et al., 2009), the validity of replacing wildfire and the ecological consequences of fire surrogate programs are largely unknown in non-upland habitats.

The implementation of prescribed fire in or near riparian forests has been conservative in the western U.S. because of concerns

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about potential negative effects on ecologically sensitive habitats and taxa (Beche et al., 2005; Dwire et al., in press; Reeves et al., 2006). The decision to introduce fire into a catchment depends on the characteristics of the catchment and the management objectives and concerns. In catchments degraded by anthropogenic disturbance, research shows that fire effects can be more severe and persistent, as ecosystem processes may not function properly (Minshall, 2003; Neville et al., 2009). In these cases, or cases involving endangered or threatened populations, managers often pursue a "no action" strategy of fire-use in watershed management. However, because many of these ecosystems have evolved with wildfire, fire exclusion from these habitats could potentially be a factor contributing to species endangerment. In intact (low anthropogenic disturbance) ecosystems, many species tend to benefit or recover quickly following wildfire disturbance (Minshall, 2003; Heck, 2007; Pettit and Naiman, 2007; Rieman and Clayton, 1997). Thus, to provide an effective surrogate for wildfire disturbance in stream systems with otherwise low anthropogenic disturbance, prescribed fire may need to alter riparian and stream communities by creating a mosaic of burned and unburned upland and riparian habitat in a manner and frequency consistent with historic, local fire regimes.

In principle, periodic prescribed fire in temperate riparian forests could result in a pseudo-natural fire regime, restoring habitat mosaics and fire-dependent communities. However, this is mostly an untested hypothesis. Many factors such as the ecological consequences of early or late season prescribed fires are poorly understood (Tiedemann et al., 2000). The lack of data on the ecological effects of prescribed fires on stream and riparian ecosystems is due to the difficulty of coordinating research with management activities and the assumption that prescribed fire effects are similar to those of wildfires. Only two studies have examined the effects of prescribed fire on stream ecosystems (Beche et al., 2005, conducted in the Sierra Nevada, CA, U.S.; Britton, 1991, conducted in South Africa), neither of which were conducted in ponderosa pine (*Pinus ponderosa*) or Douglas-fir (*Pseudotsuga menziesii*) forests of the Northern Rocky Mountains (U.S.) where thousands of hectares are treated with prescribed fires annually (Bisson et al., 2003; Friederichi, 2003). The lack of information on the effects of prescribed fire on stream habitats and species is problematic for evaluating potential impacts and benefits of fire management practices.

The goals of this study were to determine: (1) the biotic and abiotic effects of a prescribed fire on stream communities within a ponderosa pine forest, (2) if the burn severity of this prescribed fire was characteristic of others conducted in ponderosa pine forests throughout the western U.S., (3) if the burn severity and extent patterns of this and two nearby prescribed fires were similar to patterns observed following wildfire and (4) if this prescribed fire was an effective ecological surrogate for the functional role of wildfire in the stream ecosystem. To address these goals, we monitored a stream for three years before and three years after a large prescribed fire was implemented and compared biotic and habitat trends to those of four unburned reference streams over the same time period. The burn severity characteristics of this prescribed fire were compared to those of six other prescribed fires in three western states. Catchment-level burn severity and extent patterns of three prescribed fires were compared to those of wildfires using satellite imagery to quantify burn severity in riparian and upland vegetation. Prescribed fire effects on stream habitat and communities were evaluated relative to wildfire effects observed in nearby streams.

2. Methods

2.1. Study area

This study was conducted in the Payette National Forest (PNF), ID, USA (44°57′N, 115°41′W) in the Salmon River Mountains from 2001 to 2006 (Fig. 1). The five study streams were located near the confluence of the South Fork Salmon River and the East Fork of the South Fork Salmon River (hereafter South Fork). Sampled reaches ranged in elevation from 1288 to 1607 m and catchment elevations reached 2800 m. The greatest linear distance between any two streams was 23 km.

Catchments within the South Fork drainage are characterized by steep, rugged topography, and soils derived from erosion-prone Idaho Batholith granite (Hyndman, 1983). The study streams were low order (2–3) and high gradient (Table 1), and flowed through upland forests dominated by open stands of ponderosa pine at lower elevations, and mixed pine/fir (*Abies* sp.) and Engelmann spruce (*Picea engelmannii*) forests at higher elevations. Riparian communities were dominated by alder (*Alnus* sp.) and to a lesser extent, dogwood (*Cornus sericea*) and hawthorn (*Crataegus* sp.). The alders formed an upper canopy one to 15 m above ground and the understory was densely vegetated with shrubs and forbs.

Streamflow patterns in the study area were variable, depending on snowpack and timing of snow melt. During the study period, the average annual precipitation at the study area was 53 cm (data source: National Climate Data Center). Peak runoff occurred in late May or early June and, on the South Fork, averaged $100.5 \pm 47.1\ m^3\ s^{-1}$ (mean ± 1SD) over the study period (U.S. Geological Survey; Streamgage 13310700; Fig. 2). River levels return to a base flow of $4.6\ m^3\ s^{-1}$ by early August.

An active fire suppression program was initiated in 1948 and has since altered the fire regime in South Fork catchments. Moderate to large, mixed-severity (high, low, and unburned patches) fires occurred in both upland and riparian forests every 10 years on average from 1471 to 1948 (Barrett, 2000). Since 1948 the fire-free interval in most catchments has been eight times longer than pre-suppression (Barrett, 2000). The study catchments have had little other anthropogenic disturbance (i.e., few roads and little to no timber harvest or grazing activity) in the past 40–50 years.

Table 1
Catchment characteristics for streams studied from 2001[a] to 2006 within the South Fork Salmon River drainage.

Stream	Order	Mean wetted width (m)	Mean elevation (m)	Mean catchment slope (%)	Reach gradient (%)	Aspect	Mean water temperature[c] (°C)
Parks[b]	2	3.67	2181	21.2	15.2	S	12.1
Reegan	2	4.01	2092	20.6	14.3	S	12.3
Buckhorn	3	4.67	2001	20.6	5.9	SE	13.0
Blackmare	2	2.76	2159	24.9	13.2	SE	11.6
Fourmile	2	4.30	2167	23.8	10.6	W	12.3

[a] Macroinvertebrates, water chemistry, and degree-hours were first sampled in 2002. Periphyton was first sampled in 2004.
[b] Treated with prescribed fire 8 May 2004.
[c] Mean water temperature recorded at lowest 1 km reach of each stream at time of sampling over all years.

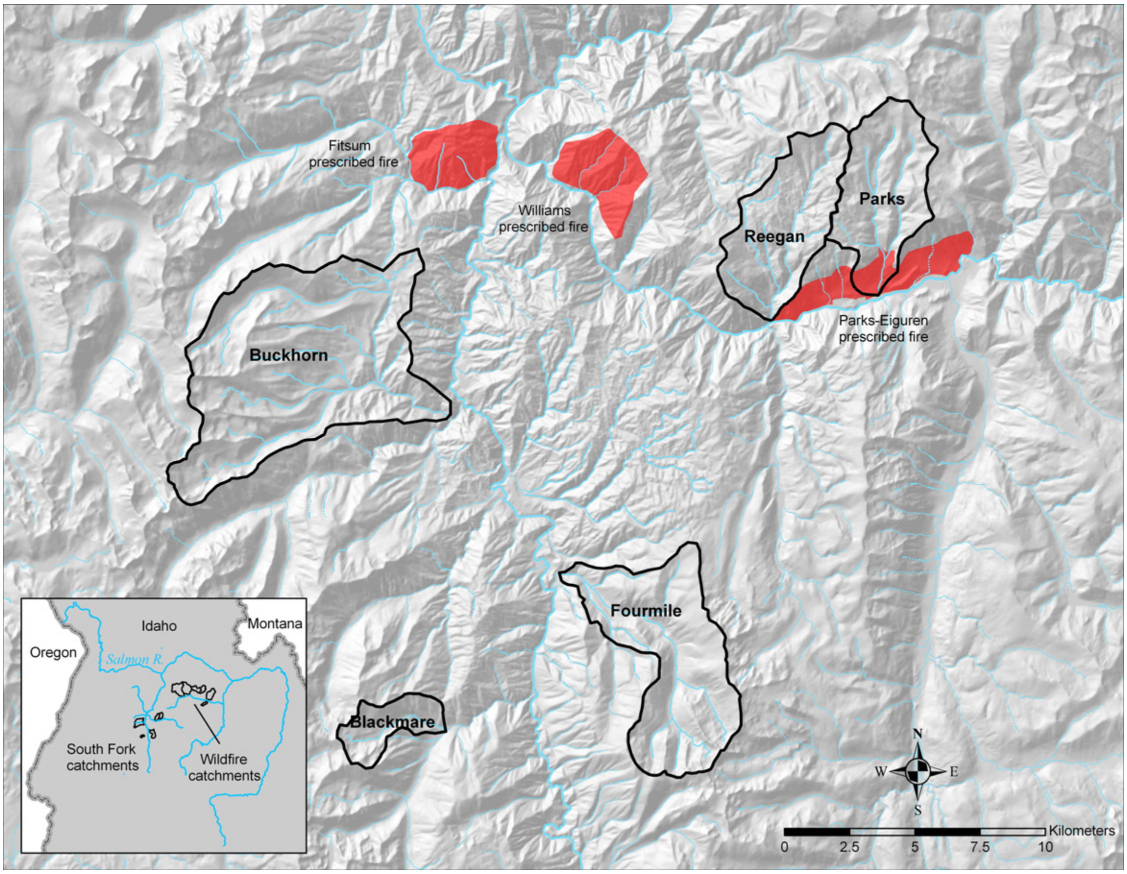

Fig. 1. Map of study catchments within South Fork Salmon River drainage, Payette National Forest, Idaho. The Parks-Eiguren, Fitsum Creek, and Williams Creek prescribed burn areas are shaded. Inset shows South Fork and wildfire burned catchments, which burned in 2000 during the Diamond Peak Wildfire and were sampled by Arkle et al. (2009) from 2001 to 2004.

2.2. Prescribed fire treatment and burn severity analyses

The Parks-Eiguren prescribed burn (2626 ha) was initiated on 8 May 2004 and burned 12% of the Parks Creek catchment as part of the Yellow Pine and Eiguren Hazardous Fuels Reduction Project (Fig. 1). The early-season burn date was chosen because higher fuel moisture leads to low intensity upland fire, and even lower intensity or no riparian fire. This strategy was selected to attain the management goals of reducing risk of crown and uncharacteristic wildfire, risk to property, and risk to natural resources (USDA Payette National Forest 2006). Aerial ignition was initiated in mixed pine/fir upland areas using incendiary objects dropped from a helicopter. Drip torches were used for manual ignition near the burn perimeter. Although the fire was not actively ignited in the riparian forest, it was permitted to burn into the riparian zone. Following the fire, aerial and ground surveys performed by U.S. Forest Service soils crews reported some patches of large tree mortality and higher burn severity (Zungia and Weaver, 2004).

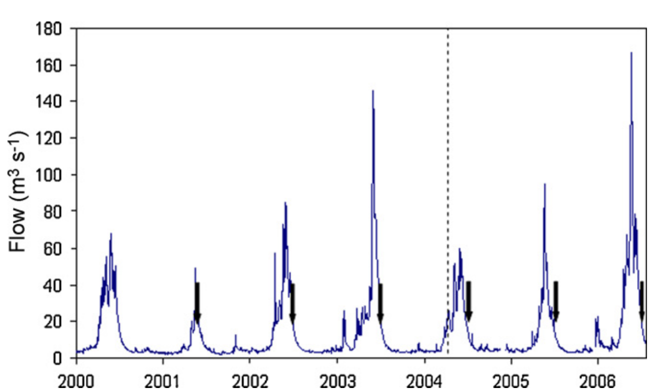

Fig. 2. Mean daily flow (m^3 s^{-1}) from 2000 to 2006, on the South Fork Salmon River at Krassel, Idaho streamgage. Arrows indicate beginning of annual sampling period. The Parks-Eiguren prescribed fire was ignited 8 May 2004 (dashed line).

We used data collected by Saab and Block (2006) to compare the burn severity of the Parks-Eiguren prescribed fire to burn severities of six other prescribed fires conducted around the western U.S. during the same time period. The six additional sites were selected, prior to prescribed burning, based on the criteria that all fires were scheduled to be carried out by the USDA Forest Service from 2003 to 2004 in ponderosa pine dominated forests (Saab and Block, 2006). Sites in Washington (Okanogan-Wenatchee N.F.), Idaho (Payette N.F.), Arizona (Apache-Sitgreaves, Coconino, and Kaibab N.F.), and New Mexico (Gila N.F.) were sampled before and after burning. Elevations at these sites were approximately 660 m in Washington, 1970 m in Idaho, 2060 m in Arizona, and 2480 m New Mexico.

Saab and Block (2006) used a modified Composite Burn Index (CBI; Key and Benson, 2006) to quantify the magnitude of fire effects (burn severity) within each prescribed fire. CBI values for understory, overstory, and overall (understory and overstory) burn severity of each prescribed fire were determined by averaging CBI values from 20 to 40 randomly located 0.4-ha circular plots per fire. The CBI value for each plot was determined by measurements of 18 variables including tree scorch-height, substrate char classification, and changes in litter, downed wood, and vegetative cover (complete list of variables in Table 3). Each variable was assigned a CBI score, ranging from 0 to 3, based on burn severity breakpoints modified from Key and Benson (2006) and an average score was calculated for each plot. Scores from 0 to 0.40 indicate no fire effect; scores from 0.41 to 1.3 reflect low burn severity, 1.31–2.3 moderate burn severity, and scores from 2.31 to 3.0 represent areas of high burn severity (modified from Key and Benson, 2006). CBI scores were calculated for 20 plots within the Parks Creek catchment of the Parks-Eiguren burn area and compared to 174 plots in the six other prescribed fire areas. If mean values for understory, overstory, and overall CBI of the Parks-Eiguren prescribed fire fell within the 95% confidence interval for the mean scores of the other six sites, then we concluded that the Parks-Eiguren prescribed fire had burn severities similar to those of the other six sites.

Catchment-level burn severity and extent patterns of the Parks-Eiguren burn and two additional prescribed fires conducted in the South Fork drainage were compared to catchment-level burn severity patterns following the nearby (~35 km to the east) Diamond Peak wildfire complex of 2000 (for study area details, see Arkle et al., 2009). We determined the percentage of the riparian zone (<20 m from a stream) and upland vegetation (>20 m from a stream) that burned at high severity and low severity for each burned catchment (both prescribed fire and wildfire), using changes in normalized burn ratios (ΔNBR) derived from pre- and post-fire Landsat 7 Enhanced Thematic Mapper Satellite imagery as described in Arkle et al. (2009). The Fitsum Creek prescribed fire (503 ha burned; 18% of catchment) and Williams Creek prescribed fire (74 ha burned; 28% of catchment) were conducted by the PNF in early May 2006 using the same methods, guidelines, and goals employed in the 2004 Parks-Eiguren prescribed fire. A description of the Diamond Peak wildfire complex is given in Arkle et al. (2009). For each catchment burned by wildfire ($n = 16$) and by prescribed fire ($n = 3$), we regressed the percentage of the entire catchment that burned against the percentage of the riparian forest that burned. For each catchment, we also compared the percentage of the entire catchment burned to the percentage of the riparian forest that burned at high severity. If these relationships were similar for catchments burned by wildfire and by prescribed fire, we concluded that the burn severity and extent patterns of the three prescribed fires were not different from those of the Diamond Peak wildfire complex.

2.3. Stream habitat sampling

We selected four catchments nearest to Parks Creek that were similar in stream width, gradient, and disturbance history to represent unburned references. Sampling in the lowest 1 km portion of each catchment ensured that our sampling reach was downstream or within the prescribed fire area of the treatment catchment and was at similar elevations among catchments. Each year (July through August 2001–2006) streams in all five catchments were surveyed using 30 randomly spaced, 1-m belt-transects (running perpendicular to the current) within the downstream-most 1-km reach of each stream. Belt-transects that occurred in waterfalls, pools, and log jams were moved a random distance upstream because they could not be sampled using our methodology. At each belt-transect, we recorded water temperature, wetted width, average depth, maximum current velocity, substrate size, and substrate embeddedness. To characterize substrate size and embeddedness, we performed a Wolman Pebble Count (Wolman, 1954) on 30 pebbles per belt-transect (900 pebbles per stream annually) using a modified substrate size class (Platts et al., 1983). The embeddedness of each pebble was categorized as: 0–5, 6–25, 26–50, 51–75, or 76–100% embedded in fine sediment. We measured the percent of each belt-transect occupied by large woody debris (LWD; >5 cm diameter) and organic debris (wood <5 cm diameter and leaf litter). We also recorded the percent of each belt-transect covered by understory riparian vegetation (<1 m from water surface), by overstory upland or riparian vegetation (>1 m from water surface), and by an undercut bank.

We measured V*, the ratio of a pool's fine sediment volume to the total volume of the pool (sediment volume + water volume), to assess mobile sediment deposition in the first 10 pools in each 1-km reach in each stream annually (Hilton and Lisle, 1993). Sediment depth was measured at five locations within a pool by pushing a metal probe through sediment until rock substrate was reached and measuring the depth from rock substrate to the interface between sediment and water. Water depth, from surface to sediment, was measured in five locations and maximum pool width and length were measured, allowing for calculations of sediment, water, and total pool volumes.

Water samples were collected for chemical analysis following spring peak flow conditions (last week in June) at each stream from 2002 to 2006. We field-processed samples (a portion of samples were collected unfiltered, unfiltered and acidified, or filtered) and shipped them overnight—in cold (4 °C) coolers to the U.S. Geological Survey's National Water Quality Laboratory. Samples were analyzed using American Public Health Association (APHA) analytical techniques for oligotrophic water chemistry (APHA, 1995).

Prescribed fire induced changes in water temperature were examined using temperature data loggers (32K StowAway Tidbit −5 to 37 °C logger, Onset Computer Corp., Bourne, MA) placed in study streams. These instruments recorded hourly water temperature on the stream bottom at one location in each study reach. We analyzed the effects of post-fire conditions on instream temperature regime using a degree-hour approach. We recorded the number of hours each year, from 2002 to 2006, where water temperatures were 16 °C or greater. This temperature was chosen because pre-fire temperatures in streams reached 16 °C on only the warmest days of summer and because this temperature is likely to be stressful for cold stenothermic taxa.

A solar radiation analysis was performed at five sampling locations in each stream to account for differences in primary production due to light availability within streams. We used Solar Pathfinder (2006) equipment (a reflective, gridded dome) and digital photography to capture an image providing year-round, site specific solar data. Images were analyzed using Solar Pathfinder Assistant version 1.1.5, which provided a monthly average kW h^{-1} m^2 based on latitude, long-term weather patterns, and site-specific vegetative and topographic shading.

2.4. Periphyton sampling

From 2004 to 2006, at each macroinvertebrate sampling location (see Section 2.5), we sampled three randomly chosen flat rocks (15 rocks per stream, annually) to determine periphyton chlorophyll-a concentration and biomass, as measured by ash-free dry mass (AFDM). Sampling and analysis followed methods from Davis et al. (2001). Autotrophic index (AI) values were calculated as the ratio of AFDM to chlorophyll-a. Typically, AI values in streams range from 50 to 200, with larger values indicating systems dominated by a heterotrophic periphyton community (APHA, 1995).

2.5. Benthic macroinvertebrate sampling

To quantify benthic macroinvertebrate communities and their habitats, we sampled five transects placed at 50 m intervals within the first 200 m of the study reach of each stream from 2002 to 2006. At each transect, we recorded the water temperature, wetted width, average depth, percent canopy cover, substrate size, and substrate embeddedness. We determined the percent of each microhabitat type (cascade, high gradient riffle, low gradient riffle, run, glide, and pool) and counted the number of LWD within each 200-m reach. Surber samples (0.10 m^2, 500 μm mesh) were collected in the thalweg of each transect during summer base flow (late August to early September). Each sample was preserved in 75–100% ethanol for identification in the laboratory. Benthic macroinvertebrates were counted and keyed to genus (family for Chironomidae and order for Oligochaeta) using Merritt and Cummins (1996). We calculated total density (all individuals), density of each genus, percent Ephemeroptera, Plecoptera, and Trichoptera (EPT), and taxonomic richness for each sample. The macroinvertebrate metrics were then averaged across the five samples taken from each stream annually.

2.6. Amphibian sampling

We sampled for Rocky Mountain tailed frog (*Ascaphus montanus*) tadpoles and Idaho giant salamander (*Dicamptodon aterrimus*) larvae and paedomorphs using kick-sampling in 30 1-m belt-transects per stream annually. All large rocks and LWD were inspected for amphibians and removed from the belt-transect area. The remaining substrate within the belt-transect was dislodged to a depth of about 10-cm. Displaced tadpoles were collected in D-frame nets at the downstream edge of the belt-transect. For each stream, we calculated an average annual density from 30 belt-transects sampled each year. These organisms provide good indicators of stream habitat conditions as they have long larval life stages (at least three years), and are known to be sensitive to sedimentation, high water temperatures, and LWD induced pool formation (Corn and Bury, 1989; Pilliod et al., 2003; Welsh and Ollivier, 1998).

2.7. Fish sampling

We recorded the presence of fish at each belt-transect either through capture during amphibian sampling or through ocular observation. We calculated fish occupancy per stream by dividing the number of belt-transects where fish were detected by the total number of belt-transects sampled per year ($n = 30$). Occupancy estimates were naïve with no adjustments for imperfect detection. We did not differentiate between fish species which included chinook salmon, *Oncorhynchus tsawytscha;* steelhead/rainbow trout, *O. mykiss* spp; westslope cutthroat, *O. clarki lewisi*; bull trout, *Salvelinus confluentus*; or sculpin, *Cottus* spp.

2.8. Stream data analysis

Stream habitat, macroinvertebrate summary metrics (density, EPT, richness), and amphibian data were analyzed using a beyond-BACI design (Before–After-Control-Impact). We analyzed each variable separately using a spatially ($n = 5$ streams) and temporally ($n = 6$ years) replicated analysis of variance (ANOVA; Underwood, 1993, 1994). This analysis technique is based on repeated measures ANOVA and is capable of detecting three types of disturbance effects (pulse, press, and increased or decreased variance) on habitat and amphibian density metrics. Independent variables used in the model for each response variable include Period (two groups: Before fire, After fire), Treatment (two groups: Reference catchments, Burned catchment), Year, Period by Treatment, and Year within Period by Treatment. Sources of variance were repartitioned and a sequence of *F*-tests allowed us to detect: 1) the three types of disturbance effects and 2) short-term (where Period is After fire only) and long-term (where Period is Before and After fire) interactions between a particular response variable and time in reference catchments (Underwood, 1993). Short-term or long-term interactions indicate that, among the reference catchments, the level of a particular response variable fluctuates asynchronously through time. For example, sediment levels in one reference stream may increase through time, while decreasing through time in the other reference streams. Except where noted, all of our analyses were performed using SAS version 9.0.1 (SAS Institute Inc., Cary, NC, USA).

Since there was no within-year replication at each catchment, a beyond-BACI analysis could not be performed on water chemistry, fish occupancy, or temperature metrics. Instead, we analyzed these data using a mean difference approach. For a given response variable, we found the average difference between values measured in unburned catchments ($n = 4$) before (2001–2003) and after (2004–2006) the prescribed fire. We then compared that value to the difference between before and after values in the prescribed fire catchment. If the before-to-after change in the burned catchment fell outside of the 95% confidence interval for the average before-to-after difference in the reference catchment, then we concluded that the fire had an effect on this variable (Beche et al., 2005).

Since no periphyton data were collected before the prescribed fire, we used general linear modeling (GLM) to compare periphyton metrics between catchments after statistically eliminating effects of light availability.

Macroinvertebrate community composition changes and relationships to habitat and treatment variables were analyzed using nonmetric multidimensional scaling (NMS) ordinations (Kruskal, 1964; Mather, 1976; McCune and Grace, 2002). Using PC-ORD 5.10 software (McCune and Mefford, 2006), we performed two NMS ordinations on the log $(x + 1)$ transformed density of individuals of each genus. In the first analysis, all five samples from each year of sampling (2002–2006) in Parks Creek were analyzed and a biplot was created relating year of sampling, annual peak flow on the South Fork streamgage (Flow), and Period to community composition. Analyzing samples from Parks Creek separately allows us to determine the relative importance of the prescribed fire, year of sampling, and annual variations in peak streamflow in explaining between-year variations in the macroinvertebrate community composition of Parks Creek. In the second analysis, we used the average density of each taxon from the five samples collected in each stream within each year ($n = 25$ sample units for this analysis; 5 streams $\times$ 5 years). We plotted these community centroids to show the similarity between the macroinvertebrate communities in Parks Creek and reference streams before and after the prescribed fire.

In both analyses, we used Sorenson (Bray-Curtis) distance to measure dissimilarity between sample units based on 250 runs of real data and 250 runs of randomized data (to provide a Monte Carlo test of each axis). Each run had a random starting configuration and provided a step-down, six axis solution. The best solution for each axis (dimension) was the solution with the lowest final stress (difference between the distance in original data and ordination data) from a run using real data. Starting from one axis, the final number of axes (dimensions) was determined by adding an axis if it reduced the stress in the final solution by ≥ 5 (on a 0–100 scale) and if that dimensionality provided significantly lower stress than the randomized runs ($p \leq 0.05$). The proportion of variance represented by each of the final dimensions was evaluated based on the correlation coefficient (r^2) between Sorenson distance in ordination space and original space. We examined the relative strength of linear relationships between community composition and treatment or environmental variables using correlations between these variables and ordination axes. In both analyses, genera present in <5% of samples were excluded, as rare taxa can have undue influence on ordination results (McCune and Grace, 2002). Additionally, sample units outside the second standard deviation for average distance

(Sorenson distance) to all other sample units were excluded as multivariate outliers (McCune and Grace, 2002).

To test for significant multivariate (density of individuals in each genus) differences between macroinvertebrate community samples collected from before or after the prescribed fire in burned or reference catchments, samples from each stream were partitioned into the following four groups: Before/Reference, After/Reference, Before/Burned, and After/Burned. We tested for significant differences between these four groups using Multi-Response Permutation Procedure (MRPP) with Sorenson distance in PC-ORD 5.06 (Biondini et al., 1985; Mielke and Berry, 2001). Unlike MANOVA and related methods, this technique provides a nonparametric analysis that does not assume multivariate normality, linearity, and homogeneity of variances. MRPP provides a test statistic (T), with more negative values indicating greater multivariate differences between groups, an effect size statistic, A (0–1), with values of one indicating perfect homogeneity within groups and 0 indicating that within-group homogeneity is no stronger than expected by chance, and a p-value indicating whether the mean within-group distance is smaller than the distance expected from chance alone (McCune and Grace, 2002).

Differences between prescribed fire and wildfire effects on stream habitat and communities were examined by comparing the results obtained here to the results of Arkle et al. (2009). All data used in these two studies were collected between 2001 and 2006 using identical methods.

3. Results

3.1. Stream habitat

We found no effects of the prescribed fire on instream or riparian habitat variables along Parks Creek. However, for several response metrics, we detected either short-term or long-term temporal interactions among the reference catchments (Table 2). No significant changes in water chemistry resulted from the prescribed fire (Table 2). In addition to those variables listed in Table 2, changes in calcium, chloride, magnesium, nitrate, potassium, sodium, and sulfate concentrations in Parks Creek were not outside of the 95% CI for Before–After prescribed fire changes in reference streams. We found that concentrations of ions and compounds in the water were very low overall. The largest Before–After fire changes were observed in conductivity and dissolved organic carbon, however these changes were of similar magnitude in burned and reference catchments (Table 2). The number of hours each year where water temperatures were $>16\,^{\circ}$C decreased more in reference streams than in Parks Creek, although not significantly (Table 2). We observed a 30% decrease in the number of degree-hours from before to after treatment in both the burned and reference streams.

3.2. Periphyton

Periphyton communities in the study streams were dominated by heterotrophs (AI > 364), as AI values fell well outside the typical range for streams (50–200). We found that AI values from Parks Creek were not significantly different from those of any reference stream in the three years post-fire after accounting for differences in solar radiation (Table 2). Parks Creek had significantly greater solar energy input during the summer months (June–August) than the reference streams (GLM, $F = 41.05$, d.f. = 24, $p = 0.0112$). The difference in light availability was likely due to topography, rather than fire effects, because the prescribed fire had no effect on riparian stream cover (Table 2).

3.3. Benthic macroinvertebrates

The prescribed fire treatment had no effect on total density, percent EPT, or genera richness of benthic macroinvertebrates in

Table 2
Summary of beyond-BACI or mean difference results for response variables averaged for 3 years before and 3 years after May 2004 prescribed fire (mean ± 95% CI).

Variable	Reference catchments (n = 4)			Burned catchment			Prescribed fire effect	Interaction among Refs.[c]
	Before	After	Δ	Before	After	Δ		
Periphyton (AI)	na	364 ± 117	na	na	411 ± 74.0	na	Unlikely	na
Macroinvertebrate:								
Density (ind/m^2)	126.5 ± 39.5	106.3 ± 28.6	−19.8 ± 51.2	220.5 ± 59.8	186.8 ± 46.5	−33.7	None[a]	Short-term
EPT (%)	54.8 ± 13.5	65.3 ± 5.97	10.5 ± 15.2	34.1 ± 33.1	58.0 ± 17.3	23.9	None[a]	Long-term
Genera richness	16.5 ± 3.37	14.8 ± 2.14	−1.78 ± 2.81	21.8 ± 3.13	18.3 ± 6.90	−3.53	None[a]	Long-term
Tailed frog density (ind/m^2)	2.7 ± 0.75	1.6 ± 0.95	−1.05 ± 0.77	2.30 ± 1.57	1.14 ± 1.46	−1.16	None[a]	Long-term
Salamander density (ind/m^2)	0.03 ± 0.02	0.02 ± 0.01	−0.007 ± 0.01	0.00	0.00	0.00	na	No
Fish occupancy (% trans.)	8.9 ± 4.9	8.8 ± 5.4	−0.01 ± 4.1	5.7 ± 2.1	8.8 ± 2.3	3.1	None[b]	na
Average pebble size (mm)	111.9 ± 7.95	118.4 ± 7.74	7.74 ± 13.1	129.8 ± 13.9	127.7 ± 18.8	2.1	None[a]	Short-term Long-term
Average pebble embeddedness (% buried)	20.3 ± 2.15	20.9 ± 2.71	0.60 ± 3.61	16.5 ± 5.42	18.1 ± 7.52	1.6	None[a]	Short-term Long-term
High riparian cover (%)	31.8 ± 7.89	29.5 ± 8.26	2.79 ± 2.07	14.2 ± 7.68	13.27 ± 1.34	−0.93	None[a]	No
Low riparian cover (%)	11.6 ± 2.81	9.63 ± 4.14	−1.97 ± 5.53	5.77 ± 2.0	5.29 ± 3.10	−0.47	None[a]	No
Undercut bank (% coverage)	6.25 ± 3.51	7.22 ± 4.43	0.97 ± 2.92	4.08 ± 4.09	5.02 ± 3.97	0.94	None[a]	Short-term
Organic debris (% coverage)	2.44 ± 1.41	2.55 ± 1.44	0.11 ± 0.52	2.01 ± 2.43	2.28 ± 2.48	0.27	None[a]	Short-term
LWD (% coverage)	2.36 ± 0.68	2.23 ± 0.79	−0.13 ± 0.37	1.44 ± 0.70	1.66 ± 0.91	0.22	None[a]	No
V*	0.07 ± 0.01	0.07 ± 0.01	<0.00 ± 0.02	0.06 ± 0.03	0.07 ± 0.01	0.01	None[a]	No
Temperature > 16 °C (h/yr)	39.8 ± 23.1	28.2 ± 19.6	−11.6 ± 15.6	8.00 ± 11.7	5.67 ± 11.1	−2.33	None[b]	na
Total P (mg/L)	0.02 ± 0.02	0.01 ± 0.01	−0.01 ± 0.02	0.015 ± 0.02	0.01 ± 0.01	−0.005	None[b]	na
Total N (mg/L)	0.22 ± 0.18	0.17 ± 0.01	−0.05 ± 0.15	0.19 ± 0.17	0.14 ± 0.01	−0.05	None[b]	na
DOC (mg/L)	1.46 ± 0.47	1.63 ± 0.45	0.17 ± 0.64	0.95 ± 0.29	1.13 ± 0.09	0.18	None[b]	na
Conductivity (uS/cm)	37.9 ± 11.0	34.8 ± 6.1	−6.38 ± 2.69	35.9 ± 10.3	29.7 ± 3.9	−6.28	None[b]	na
pH	7.52 ± 0.13	7.35 ± 0.11	−0.21 ± 0.07	7.56 ± 0.11	7.42 ± 0.10	−0.13	None[b]	na

[a] Beyond-BACI asymmetrical ANOVA sequence of tests used to detect impacts
[b] Mean difference approach used to detect impacts, which occur when burned catchment difference is outside 95% CI of the before-to-after difference in the reference catchments.
[c] Interactive system based on Underwood (1993). Short-term indicates whether, for a particular variable, reference catchments have different short-term (after prescribed fire) trends from one another. Long-term indicates whether reference catchments have different before-to-after prescribed fire trends. In a noninteractive system (indicated by "No"), reference catchments have similar short and long-term trends to one another.

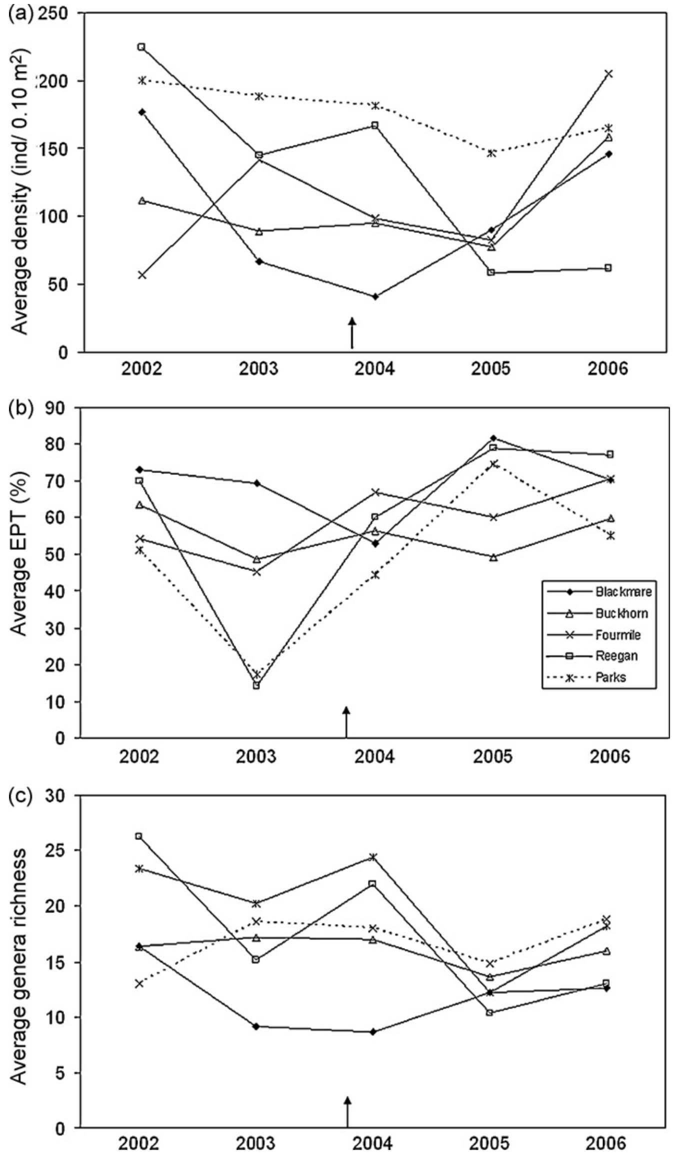

Fig. 3. Average macroinvertebrate (a) density, (b) percent EPT, and (c) genera richness in reference (solid lines) and burned (dashed line) catchments. Error bars omitted for clarity. Arrow in each panel indicates prescribed fire date.

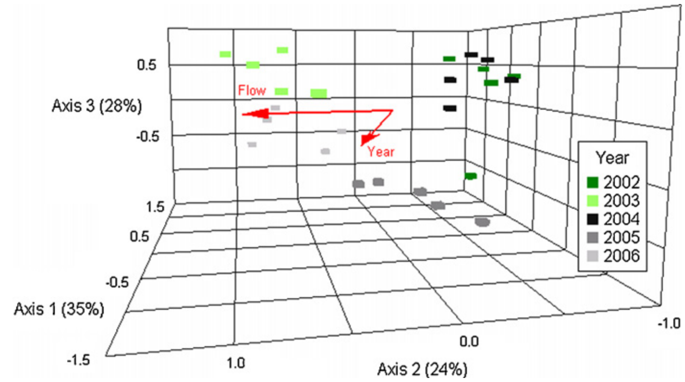

Fig. 4. NMS ordination of five Surber samples per year in three-dimensional genera space for Parks Creek. Samples closer together are more similar based on community composition. The percent of variance represented by each axis is included in parentheses. Annual peak flow is positively related to Axis 3 (r^2 = 0.78), Year is negatively related to Axis 3 (r^2 = 0.34), and Period is below the cutoff value (r^2 = 0.20) with all axes.

the study stream (Table 2; Fig. 3a–c). Likewise, we found that a NMS ordination of all five samples from each year of sampling in Parks Creek showed no fire effect. A three axis solution (stress = 9.93, p = 0.0196) was chosen, representing 87.8% of the variance in the original data. This solution was reached after 82 iterations and resulted in a final instability of 0.00001. The ordination biplot relating Year, Flow, and Period to community macroinvertebrate composition illustrates that annual peak flow was strongly, positively correlated (r^2 = 0.78) with Axis 2, which represented more variance (38%) than the other two axes (Fig. 4). We found that year of sampling (Year) was negatively correlated (r^2 = 0.34) with Axis 3, which represented 28% of the variance between distance in original and ordination space. Period did not correlate strongly enough with any axis to reach the cutoff value (r^2 = 0.20). Individual Surber samples clustered according to Year, rather than by permanent plot location within Parks Creek (Fig. 4).

The biplot from our second NMS ordination illustrates the community similarity between the burned stream and reference streams before and after the prescribed fire (Fig. 5). A three axis solution (stress = 13.98, p = 0.004) was chosen, representing 85.6% of the variance in the original data. This solution was reached after 115 iterations and resulted in a final instability of <0.00001.

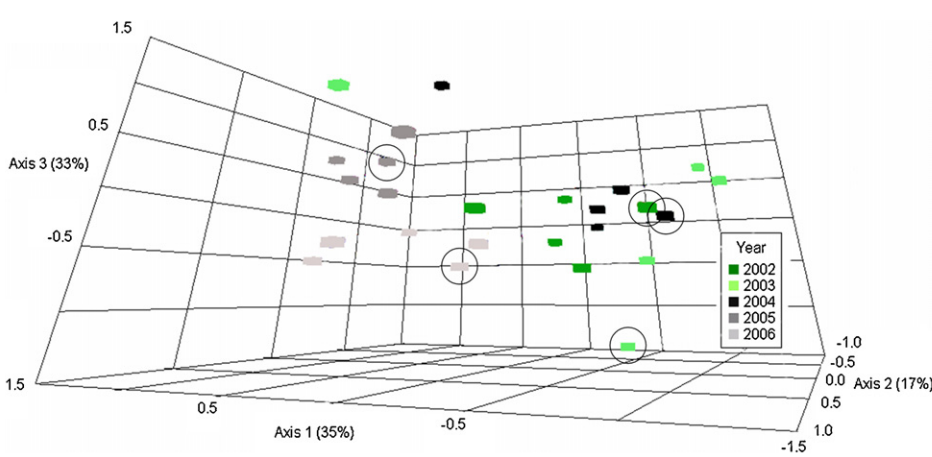

Fig. 5. NMS ordination of mean values of five samples for each stream per year. Sample unit centroids are plotted in three-dimensional genera space. The percent of variance represented by each axis is included in parentheses. Parks Creek samples from each year are circled.

Table 3
MRPP results assessing the difference in macroinvertebrate community composition between samples grouped by Period (Before/After prescribed fire) and Treatment (Reference/Burned). Each sample assigned to a group represents the five Surber sample average of the density of individuals from each genus for each stream in each year.

Groups compared	Sample size	T[a]	A[b]	p-value
Bef-Ref vs. Aft-Ref	8 vs. 12	−1.959	0.023	0.046
Bef-Red vs. Bef-Burned	8 vs. 2	1.182	−0.026	0.883
Bef-Ref vs. Aft-Burned	8 vs. 3	0.726	−0.015	0.752
Aft-Ref vs. Bef-Burned	12 vs. 2	−0.746	0.015	0.216
Aft-Ref vs. Aft-Burned	12 vs. 3	0.532	−0.009	0.671
Bef-Burned vs. Aft-Burned	2 vs. 3	0.624	−0.044	1.000

[a] More negative values of T indicate stronger separation between groups.
[b] The A statistic provides a sample size-independent description of the magnitude of difference between groups, with $A = 1$ indicating perfect within-group homogeneity and maximum effect size.

We found no prescribed fire effect based on the multivariate difference in macroinvertebrate community composition between samples grouped by Period and Treatment (Table 3). The negative value of T (−1.959) and relatively high A value (0.023) for the comparison between Before and After prescribed fire in the reference streams, indicates that the strongest separation is between these two groups. The sample size-independent A statistic for the comparison of Before to After the prescribed fire in the burned stream indicates a relatively small difference between groups and low within-group homogeneity. Community composition in reference and burned streams was also found to be similar Before (T = 1.182, A = -0.026) and After (T = 0.532, A = -0.009) the fire treatment.

3.4. Amphibians

We found that the prescribed fire had no immediate or delayed effects on the density of tailed frog tadpoles in Parks Creek. Tadpole densities decreased by 50% in both reference and burned catchments after the prescribed fire (Fig. 6). The effect of Period on tadpole density varied among reference streams (Table 2), with some reference streams exhibiting a significantly different pre-to-post-fire pattern (i.e., long-term interaction) in tadpole density than others. However, we found no significant short-term interaction in tadpole density trends among reference streams. In other words, all reference streams had similar trends in tadpole density after the fire (Fig. 6). We did not find Idaho giant salamanders in the burned catchment before or after the prescribed fire, and their densities appeared to be stable within reference catchments (Table 2).

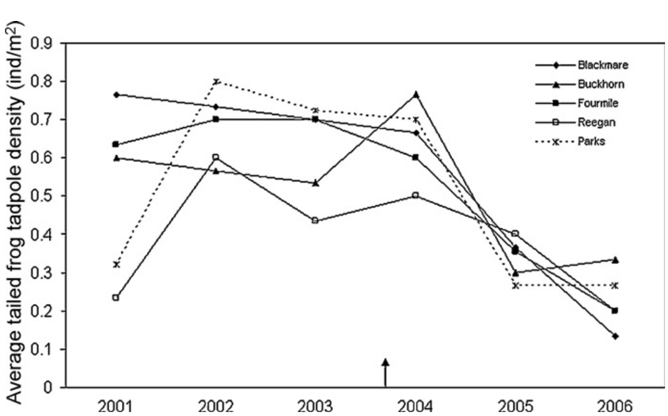

Fig. 6. Average tailed frog tadpole density from 2001 to 2006. Values represent mean density from 30 transects per stream annually. Error bars omitted for clarity. Arrow indicates prescribed fire date in Parks Creek catchment.

Table 4
Average composite burn index (CBI) scores and burn severity class (see Section 2.2) for each variable measured in 20 plots within the Parks-Eiguren prescribed fire boundary. Data from Saab and Block (2006).

Variable	Average CBI score	Burn severity
Ground char	0.63	Low
Rock char	0.35	Unburned
Litter char	0.63	Low
Log char	1.22	Low
Δ Ground cover	2.23	Moderate-High
Δ Litter cover	0.45	Low
Δ Herb cover	0.38	Unburned
Δ Shrub cover	0.63	Low
Δ Small (dbh[a] < 8 cm) tree density	1.58	Moderate
Δ Fine DW[b] (LED[c] < 8 cm) dry weight	1.52	Moderate
Δ Med DW (8 < LED < 23 cm) dry weight	1.08	Low
Δ Large DW (LED > 23 cm) dry weight	1.78	Moderate
Δ Med tree (8 < dbh < 23 cm) density	0.98	Low
Δ Large tree (dbh > 23 cm) density	0.33	Unburned
Tree char height	0.68	Low
Percent green canopy	0.43	Low
Percent brown canopy	0.40	Unburned-Low
Percent black canopy	<0.01	Unburned

[a] Diameter at breast height.
[b] Downed wood.
[c] Large-end diameter.

3.5. Fish

The average percent of belt-transects where we detected fish increased by 3.1% in Parks Creek after the prescribed fire, however this increase was not outside of the 95% CI for the average change in the reference streams, suggesting no significant effect of prescribed fire (Table 2).

3.6. Prescribed fire burn severity

Composite Burn Index (CBI) scores compiled by Saab and Block (2006) indicate that the burn severity of the Parks-Eiguren prescribed fire was in the low to moderate range for most of the 18 variables measured (Table 4). The highest CBI scores (greatest magnitude of change or fire effect) were observed for log char, change in ground cover, downed wood mass, and tree density (Table 4). CBI values for understory, overstory, and overall burn severity in the Parks-Eiguren prescribed fire fell within the 95% confidence interval of mean values from six prescribed fires conducted from 2003 to 2004 in ponderosa pine forests in Arizona, New Mexico, and Washington (Table 5).

Burn severity and extent patterns of the Parks-Eiguren, Fitsum Creek, and Williams Creek prescribed fires were substantially different from those observed following the Diamond Peak wildfire complex. Using remote sensing, we estimated that 12% of Parks Creek catchment burned (3.8% of the riparian forest burned, nearly all within the 1 km stream survey reach), 18% of Fitsum Creek catchment burned (2.0% of the riparian forest burned), and 28% of Williams Creek catchment burned (0% of the riparian forest burned)

Table 5
Average composite burn index (CBI) scores (0–3) by vegetative strata for Parks-Eiguren prescribed fire and average ± 95% CI values for six other prescribed fires in western U.S. ponderosa pine forests from 2003 to 2004. Data from Saab and Block (2006).

Strata	Parks-Eiguren (20 plots)		Mean of other fires (174 plots)	
	CBI score	Severity class	CBI score	Severity class
Understory	0.90	Low	0.87 ± 0.20	Low
Overstory	0.47	Low	0.68 ± 0.22	Low
Overall	0.85	Low	0.88 ± 0.18	Low

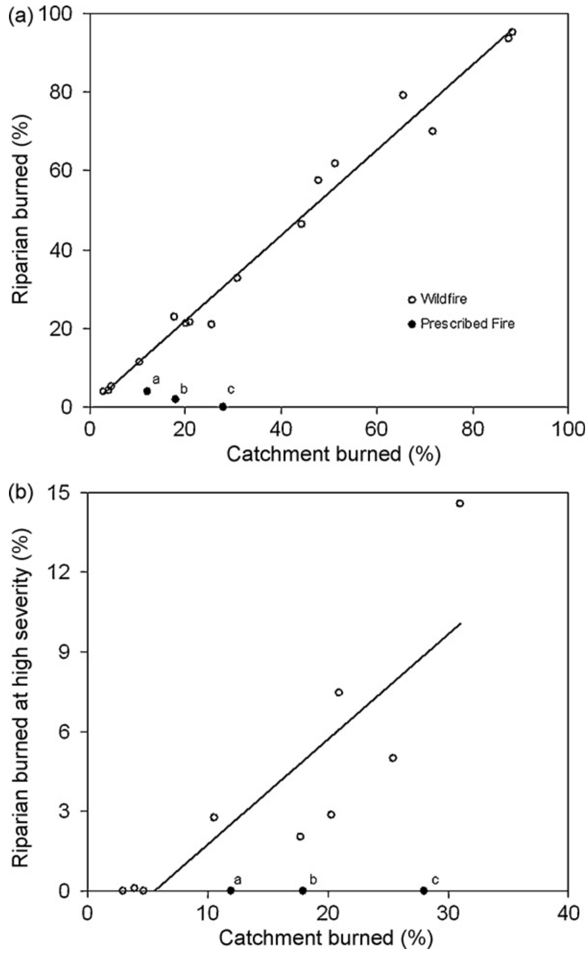

Fig. 7. (a) The percentage of each catchment's riparian forest burned against the total percentage of each catchment burned by wildfire (n = 16; open circles) and by prescribed fire (n = 3; filled circles). (b) For wildfire catchments where 30% or less of the catchment burned, the percentage of each catchment's riparian forest burned at high severity against the total percentage of each catchment burned by wildfire (n = 9) and by prescribed fire (n = 3). Parks Creek, Fitsum Creek, and Williams Creek catchments are labeled a, b, and c, respectively.

during their prescribed fires. In 16 catchments burned by wildfire, we found a significant, nearly one-to-one, positive relationship between the total percentage of a catchment burned and the percentage of the riparian forest burned (F = 800, d.f. = 15, r^2 = 0.98, $P < 0.0001$). The three prescribed fires did not follow this trend. The percentage of riparian vegetation burned by prescribed fire was less than a third of the amount expected by a wildfire of similar extent (Fig. 7a). Moreover, in catchments where less than 30% of the area burned (extents similar to the three prescribed fires), we found a significant, positive relationship between percentage of the catchment burned and percentage of the riparian forest burned at high severity in the catchments burned by wildfire (F = 18.3, d.f. = 8, r^2 = 0.72, P = 0.0036), but not in those burned by prescribed fire (Fig. 7b). None of the catchments burned by prescribed fire had any high severity fire in the riparian forest (Fig. 7b).

3.7. Wildfire versus prescribed fire effects on stream ecosystems

The effects of the Parks-Eiguren prescribed fire were not analogous to those of the Diamond Peak wildfire for any of the nine variables examined (Table 6).

4. Discussion

Despite steep topography, erosion-prone soils, and sampling directly within the burned area, we found no immediate (1–3 month) or delayed (≤3 years) effects of the prescribed fire on the biotic and abiotic characteristics of the study stream. Although a case study on one stream, these data provide some of the first empirical information on the effects of prescribed fire on stream ecosystems. As far as we know, there are only two published studies investigating this question and neither assessed the effects of prescribed fire on multiple taxonomic groups including amphibians and fish (Beche et al., 2005; Britton, 1991). This is also the first study to examine prescribed fire burn severity and extent of upland and riparian vegetation in the context of effects on stream habitat and communities.

Our findings are mostly congruent with the limited research on prescribed fire effects on riparian and aquatic ecosystems. In a low-order stream in the Sierra Nevada, Beche et al. (2005) found no prescribed fire effects on LWD and V*, but did observe short-term (≤1 year) effects on the concentrations of four ions and periphyton biomass. They observed an immediate (10–19 day) effect on macroinvertebrate community composition, but no effects on benthic macroinvertebrates thereafter. A possible explanation for these differences to our study is that the prescribed fire in their study was actively ignited within the riparian forest. In contrast, riparian vegetation was mostly unburned or burned at low severity in this prescribed fire despite no active fire exclusion from the riparian zone. In another study where prescribed fire was excluded from the riparian area, no effects were observed on stream macroinvertebrates (Britton, 1991).

Many of the variables we analyzed exhibited asymmetrical fluctuations through time in the reference streams. This indicates that to have a significant effect, the disturbance would have to be fairly intense, or perhaps burn closer to the stream, to be outside the range of annual variation among the reference streams (Underwood, 1993). Stream systems are often dynamic and

Table 6
Comparison of prescribed fire and wildfire effects on stream habitat and communities from 1 catchment treated with prescribed fire and 6 catchments burned by wildfire. All data were collected between 2001 and 2006 using the same sampling design and methods in both studies.

Variable	Prescribed fire effect[a]	Wildfire effect[b]
Macroinvertebrate community composition	None	Increased annual variability
Tailed frog density (ind/m^2)	None	Decreased[c]
Fish occupancy (% trans.)	None	Increased annual variability[c]
Riparian cover (%)	None	Decreased
V*	None	Increased annual variability
Organic debris (% coverage)	None	Increased annual variability
LWD (% coverage)	None	Increased annual variability
Undercut bank (% coverage)	None	Increased annual variability
Water temperature > 16 °C (h/yr)	None	Increased[c]

[a] Prescribed fire effects based on this study.
[b] Wildfire effects based on Arkle et al. (2009).
[c] Pilliod and Arkle (unpublished data).

heterogeneous (Minshall, 2003; Resh et al., 1988). Consequently, disturbances may have to be severe, cover a large proportion of a catchment, or impact key habitat types (e.g., riparian forests) to have a detectable effect. For example, some studies suggest that the threshold for initiation of wildfire effects on aquatic macroinvertebrates occurs when 20–50% of the catchment burned, depending on geologic and climatic conditions (Minshall, 2003).

4.1. Was the prescribed fire characteristic of others in ponderosa pine forests?

High fuel moisture present during this spring burn resulted in a low intensity ground and surface fire in the upland vegetation and little fire within the riparian vegetation. This pattern is consistent with other prescribed fires conducted in ponderosa pine forests (Saab and Block, 2006) and on the PNF. Hence, we suspect that the community and ecosystem responses that we observed may be typical for treatments conducted under similar conditions in this forest type.

The prescribed fire met the management goals of reducing ground and surface fuels and risk to nearby property, while maintaining pre-fire conditions within riparian and stream systems. This outcome may be important for other managers who are considering reintroducing fire into upland forests without changing riparian and stream conditions. Some managers may not want to change instream conditions that provide habitat for sensitive or federally listed species.

4.2. Was the prescribed fire an ecological surrogate for wildfire?

In contrast to the lack of effects following the Parks-Eiguren prescribed fire, Arkle et al. (2009) report that a nearby mixed-severity wildfire affected riparian and stream habitats and communities in comparable streams. Macroinvertebrate community composition was increasingly variable with increasing riparian burn severity for four years following wildfire (Arkle et al., 2009). Tailed frog densities were lower in the wildfire burned catchments compared to unburned catchments (Pilliod and Arkle unpublished data, also see Hossack et al., 2006). In-stream habitat components known to influence biotic communities dramatically increased (e.g., water temperature), decreased (e.g., riparian canopy cover), or exhibited increased inter-annual variability (e.g., sediment, large wood) for several years post-wildfire (Arkle et al., 2009). Even after controlling for differences in burn extent between wildfire and the prescribed fire (resulting in 2 wildfire catchments with less than 12% burn extent), Arkle et al. (2009) found effects of wildfire on instream habitats and communities. The primary difference between these two wildfire catchments and Parks Creek catchment was that in the wildfire burned catchments, the riparian forests along each stream burned proportionately to the amount of upland vegetation burned in each catchment. However, a four-fold increase in the amount of riparian forest burned by the prescribed fire would be required for the Parks-Eiguren burn to mimic the wildfire burn pattern within Parks Creek catchment. Further, some riparian vegetation burned at high severity during the wildfire, but none burned at high severity during the prescribed fire. These differences could be critical since many of the in-stream habitat components affected by the wildfire are derived from the riparian forest (e.g., canopy cover, large wood, organic debris, sediment). In conclusion, it does not appear that this prescribed fire served as an ecological surrogate for wildfire disturbance in this forest.

4.3. Management implications

The spring ignition timing of this prescribed fire and the low amount of fire in the riparian forest (also observed in two additional prescribed fires) may have important implications for forest and resource management. Despite the tendency for wildfires in this forest to burn in summer or early fall and the importance of seasonality in determining fire effects on riparian and stream habitats (Pettit and Naiman, 2007) many prescribed burning programs make extensive use of early-season fires. We recognize, and our findings support the notion, that early-season ignition is an important low-risk means for managers to accomplish needed fuel reductions and upland habitat modifications while limiting impacts on riparian and stream habitats. Although not the case here, prescribed fire is often intentionally excluded from riparian forests to avoid fire-associated increases in sediment levels and other habitat changes that could be detrimental to sensitive aquatic species (Bozek and Young, 1994; Rinne, 1996). However, regardless of whether prescribed fire is intentionally or unintentionally excluded from riparian vegetation, mixed-severity, patchy fires within riparian forests were common in many ponderosa pine forests prior to the era of active suppression policies (Barrett, 2000; Everett et al., 2003; Olson and Agee, 2005). Transmission of fire disturbance from upland to riparian vegetation was an integral component of riparian and stream systems in dry forest types in the western U.S. (Bisson et al., 2003; Everett et al., 2003; Naiman and Decamps, 1997; Reeves et al., 1995; Rieman and Clayton, 1997). Riparian fires likely increased instream habitat heterogeneity through periodic alterations of structural components (e.g., LWD), altered primary production (Knutson and Nae, 1997; Pilliod et al., 2003; Rieman and Clayton, 1997), and likely maintained biodiversity by removing stronger competitors (Dwire and Kauffman, 2003; Olson et al., 2007). A "prescribed fire regime" of repeated early-season prescribed fires that do not burn riparian forests could perpetually exclude disturbance from riparian and stream communities. The consequences of this are unknown, but disturbance exclusion could be detrimental to biodiversity in stream and riparian communities where, for millennia, species have adapted to wildfire disturbance (Agee, 1993; Bisson et al., 2003). Despite the logistical difficulties involved in such studies, further work on this topic is needed to move beyond case studies of prescribed fire effects on stream ecosystems and towards a complete understanding of the ecological implications of this management tool in different forest types.

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